Food waste management strategies: the feasibility of food waste disposers (FWD) in Australia

FINAL REPORT

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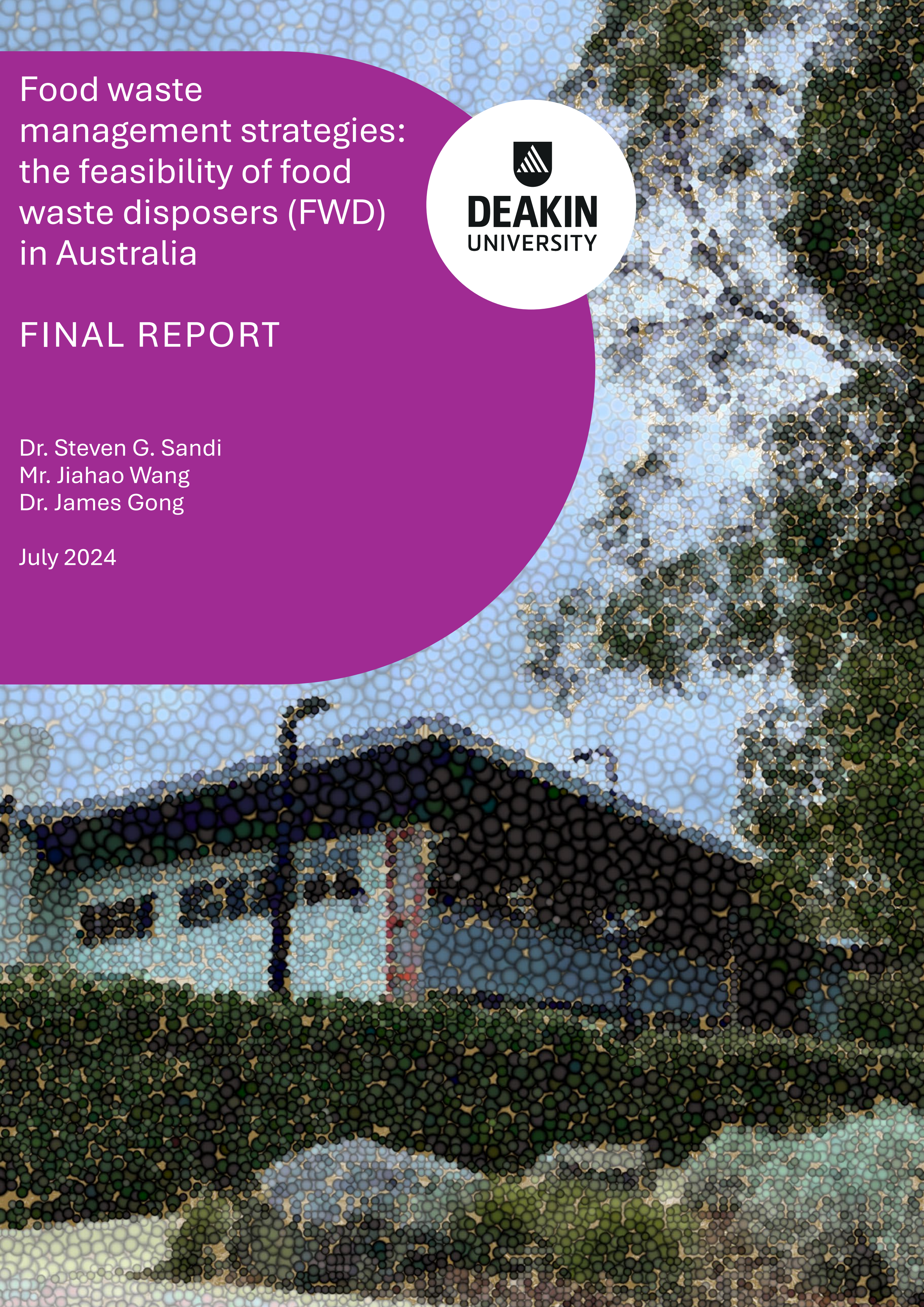

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EXECUTIVE SUMMARY

Food waste disposers (FWDs) have been recognised as a potential means to divert food waste from landfills and help achieve circular economies. The overall implementation of such devices worldwide changes from region to region, with a large penetration in the US and Canada. In Australia, it is estimated that 12% of households have such devices installed.

Abey Australia has approached Deakin University to conduct a study on the benefits of implementing FWDs in Australian households and the state of the art of food waste management in Australia. The study covers a literature review of current waste management strategies and the most common arguments in the literature regarding the use of FWDs. In addition, Abey Australia required performance testing of two different FWD devices at Deakin's laboratory facilities. The performance of the devices was determined in terms of food grinding rates and degree of fineness.

Food waste management strategies in Australia focus on implementing high-level policies encouraging the decrease of food waste at the production and retail levels, with the encouragement of food donations and research into new technologies that reduce food waste at the source. At the household level, however, 2.46 million tons (Mt) of food waste is being produced annually with 1.8 Mt of this food destined for landfills. The general strategy assumed by councils across most states is the recovery and recycling of food with the implementation of food organics / green organics bins (FOGO), however, this has not been fully adopted across all states and targets for its adoption vary between states.

The installation of FWDs is currently banned in Western Australia. In the rest of the country, their implementation is up to the individual council or entity in charge of wastewater treatment. For example, while Sydney Water does not allow the connection of FWDs to their network, Brisbane City council is promoting a rebate program that encourages users to install FWDs.

In terms of water usage from FWDs in the Australian context, an estimate of 12.4 L/kg of food waste was found in the literature, equivalent to 2.95 L/person/day. The average electricity usage is 0.02 kWh/kg of food disposed. For a three-person household, this translates into $37.7 AUD per year for operation costs. The cost of purchasing and installation of a FWD device was estimated to be between $1,055 and $1,745 AUD based on approximations of current unit prices and labour.

Common concerns regarding the impacts of FWDs on sewers were found to be unfounded based on field studies. Though some level of residues can be expected in the sewers from fat, oils and grease in the food waste, most observations have not been able to conclude there is a direct impact. The increased loadings in the wastewater treatment plants (WWTP), can increase electricity consumption for aeration as part of the wastewater process, but studies on circular economy and life cycle analysis have shown that the implementation of energy recovery technologies at the WWTP can be a feasible way to recycle organic materials and reduce greenhouse gas emissions when compared against other treatment methods like landfill and food waste composting. Results seem to be variable, with FWDs performing better than food waste composting in some scenarios and vice versa.

High-level calculations for a case study in Melbourne was carried out, determining the cost for customers to operate FWDs and the comparison of two scenarios. In Scenario 1, landfill waste disposal is done via curbside collection which is then taken to landfill. No recovery of biogas is considered in Scenario 1. In Scenario 2, use of FWDs for food waste diversion and treatment at the WWTP with biogas recovery is considered. In terms of energy required, Scenario 1 outperforms Scenario 2, while Scenario 2 has much smaller green gas emissions. Based on the results, a level of FWD penetration up to 25% is feasible in Melbourne considering current limitations in energy that can be produced at the WWTPs and planned upgrades by Melbourne Water. The analysis of these two polarizing scenarios shows an optimal solution for food waste diversion should be a combination rather than a single strategy and citizens should be presented with options they are willing to use.

The performance of two different FWD models was tested at Deakin's laboratory. The two devices are the WDU-3ASAU with a ¾ horsepower Stealth-Motor aimed at small to medium families (2 to 6 family members), and the WDU-4ASAU with a 1 ¼ horsepower Stealth-Motor aimed at larger families (more than 5 family members). The devices were tested using several mixed food loads and using the Australian Standard AS 2396-1980 Performance of Household Waste Disposal Units. The status of the standard is listed as "withdrawn" on the website of Standards Australia and no other versions were found; therefore, the methods were used mainly as a guideline for the testing. The devices achieved a grinding rate of 500 g/minute in average when using vegetables, meat, eggshells, and other "soft" waste, which translates into energy requirements of 0.019 kWh and 0.030 kWh for the WDU-3ASAU and the WDU-4ASAU devices respectively. In terms of bone grinding capacity, the devices achieved a grinding rate of 100 g/minute, resulting in energy requirements for grinding bone material five times higher than other food waste.

In the degree of fineness tests, most of the wet ground material was retained by the 2.36 mm sieve due to the congealing of fat material and capturing of other food particles. Overall, the devices met the requirements set by the standard in at least one of the food samples. It is unclear if the particle size distribution can have potential effects on the wastewater treatment process. This may require further investigations that are outside the scope of this study. Finally, using a clear pipe arrangement, some sedimentation of food waste was observed at the pipe bends; however, the tests suggest no potential clogging of pipes due to ground food waste.

1 INTRODUCTION

Municipal food waste disposal constitutes a major issue worldwide. In Australia, municipal food waste represents 35 to 45% of the total municipal waste, totalling 1.6 million tons (Mt) of food waste, 91% of which ends up in landfills (Edwards et al., 2018; Randell et al., 2014). Landfill disposal of food waste is a major contributor to greenhouse gas (GHG) emissions; therefore, food waste diversion from landfills is imperative to reduce solid waste and mitigate GHG emissions.

Food waste disposers (FWD) are household devices installed under the kitchen sink that allow for the grinding of non-edible food waste and are connected directly to the wastewater collecting system; therefore, food waste slurry resulting from the grinding process is not disposed of as solid waste and is safely diverted to a wastewater treatment plant (WWTP) for processing. Naturally, the use of FWDs as a food waste diversion method can significantly reduce the amount of solid waste sent to landfills (Marashlian & El-Fadel, 2005; Yang et al., 2010). Recent studies, testing emerging technologies for WWTPs to make the best use of additional organic waste, have shown that FWDs can be a feasible food waste diversion option (Moñino et al., 2017; Thota Radhakrishnan et al., 2018). Despite recent advances, FWDs have been questioned due to the additional energy and water consumption to operate the devices in the household as well as the impacts of increasing loadings and potential greenhouse gas emissions at the WWTP. Studies conducted worldwide have shown a diverse range of results, leading to contradictory opinions and perspectives on the use of FWDs.

A long-term study conducted in Surahammar, Sweden, showed a 39% reduction in waste sent to landfills between 1996 and 2007, and no significant increase in hydraulic, organic and chemical loadings at the WWTP (Evans et al., 2010). A small-scale study conducted in Japan showed that the implementation of FWDs decreased the amount of municipal waste reaching landfills by more than 50% (Yang et al., 2010). Estimates of food waste diversions using FWDs in the greater Beirut area of Lebanon showed potential for diverting food waste between 12% to 43% but with increases in organic loading of up to 62% (Marashlian & El-Fadel, 2005). Similarly, a study focusing on the impact of FWDs on GHG from WWTPs in Korea reported potential increases of 51.2% in the organic loading entering the WWTP if all food waste is diverted, resulting in potential increases of 58% in net GHG emissions (Kim & Phae, 2023). However, comparing GHG emissions from direct landfilling, a study of WWTP in Hong Kong and USA shows a reduction of 35% GHG emissions when FWDs are included in the municipal wastewater treatment process (Guan et al., 2022). Regarding opportunities for reducing the cost of waste collection versus impacts to the wastewater industry, a study in the UK showed that benefits from FWDs will only occur if large-scale adoption from users occurs, but if adoption is low, then FWD implementation should be regulated (Iacovidou, Ohandja, & Voulvoulis, 2012). However, it was later argued that several key considerations were not taken in the UK study (Evans, 2012), and authors advocated for a range of food waste diversion methods (including the use of FWDs) and giving users options they are willing to use, rather than promoting a single option for food waste management.

Despite the contradictions and conflicting publications on the use of FWDs, the potential use of energy recycling technologies at the WWTP to reduce emissions, such as processing biogas from wastewater and food waste, has been recognized in the USA (Shen et al., 2015), Europe (Zupančič et al., 2022) and Australia (de Haas et al., 2013). This highlights how the use of FWDs can potentially result in higher biogas production from the WWTP if the facilities are adapted to handle the excess wastewater. Davis et al. (2005), commented on several benefits from the introduction of FWD as part of municipal waste management in Europe. For example, it is argued that the additional organic content from food waste helps the biological nutrient removal, and in Italy, the Italian Ministry of Environment has recommended that water authorities provide free FWDs to residents in areas where insufficient organic material is reaching wastewater treatment facilities (Davis et al., 2005).

As shown above, there is recognized potential for implementing FWD in the municipal waste treatment process and a large range of literature. This report aims to show the state of the art in food waste management in Australia and to investigate the feasibility of implementing FWD in Australian households. The following sections present a comprehensive review of food waste management and current research on the implementation of FWDs. The report then provides an overview of the utilization of FWDs by estimating potential gas emissions in a study case within the Australian context. Finally, the report concludes with a section dedicated to testing on the performance of two different FWDs as required by Abey Australia. The testing was conducted at the laboratory facilities of Deakin University. Results and observations during the testing of the devices are provided.

2 FOOD WASTE MANAGEMENT

2.1 Food Waste Reduction Strategies

Food waste management refers to the systematic approach and strategies implemented to handle and redirect food and agricultural products for better purposes such as human consumption, animal feed, industrial application, and other environmental benefits. In recent years, several countries have started implementing strict policies to reduce food waste. In 2016, the French government passed a law that requires grocery stores to donate edible food instead of throwing it out and grocery stores managers can be fined or face jail time if caught violating this law (Chrisafis, 2016). Though this has contributed to a significant reduction of the amount of food waste from stores, each year France continues to produce 9 Mt of food waste, 46% which comes from households (EEA, 2023b). Similarly, in other European countries such as Norway and Denmark, agreements have been put in place to reduce food waste. Through campaigns and education, consumers in Norway are now more knowledgeable at reducing food waste and food donations receive tax breaks (EEA, 2021). Denmark has one of the highest waste generation rates per capita in Europe, yet the amount of food waste per week was reduced 8% between 2011 and 2017 (EEA, 2023a) through the use of several campaigns and non-profit organizations such as Stop Spil At Mad (Stop Wasting Food). Japan has a serious food supply and food waste problem due to a lack of self-sufficiency to produce as much food as required by the population. Current Japanese policies to meet the UN Sustainable Target promote the conversion of food waste form the commercial sector into feed for livestock, compost and energy recycling (Okayama & Watanabe, 2024). However, Japan's household food waste production is estimated to be 10.14 Mt, of which 94% is incinerate or sent to landfill and only 4% is recycled (Liu et al., 2016).

Food waste is also a major problem in Australia. Similarly to many other nations, the National Food Waste Strategy aims to reduce Australia's food waste by 50% (from the level in 2016 year) by the year 2030 (Commonwealth of Australia, 2017). This is a commitment to achieve the United Nations Sustainable Development Goal 12.3 of halving food waste per capita at retail and consumer levels by 2030 (United Nations General Assembly, 2015). The strategy sets out a framework of actions to meet this target, which requires a reduction of food waste across diverse parts of the supply chain, from food production and processing to retailers. Figure 1 shows the hierarchy of food waste adopted by the strategy. Avoidance or prevention of food waste before reaching consumers is the option with the highest level of preference, referring to consumer education and promoting research and development for efficient food production. It is recognized that some of the major drivers of household food waste are confusion over “use-by” and “best-before” dates, over-purchasing of food that is then thrown out, limited knowledge of how to repurpose or store food leftovers and limited access to food waste collection systems (Commonwealth of Australia, 2017). As such, education of consumers is highly important for effectively reducing food waste. Disposal of food waste directly to landfills or through sewer and wastewater treatment are considered as the least preferred options. Additionally, considerations for reuse, recycling, reprocessing and energy recovery from food waste are in between avoidance of food waste and disposal.

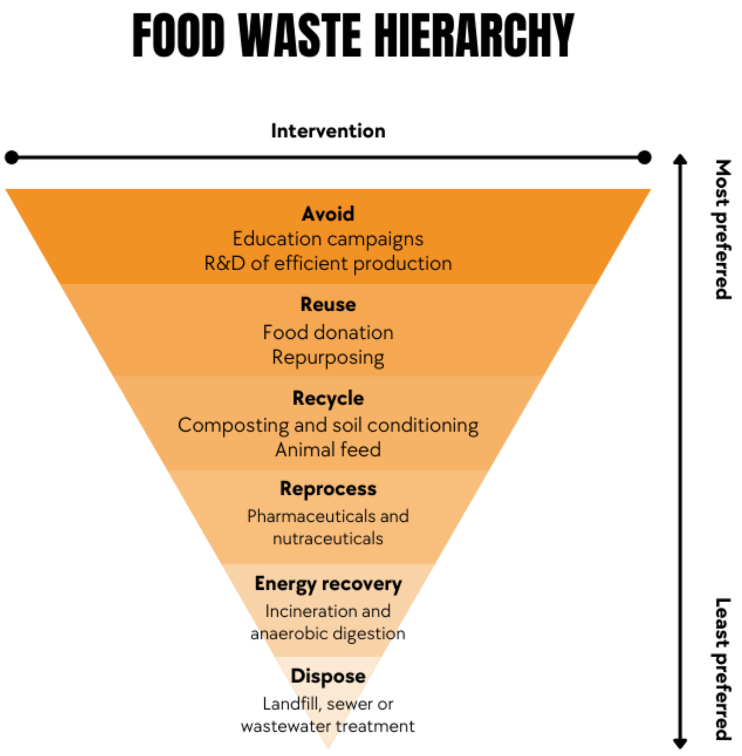

Figure 1. Food waste hierarchy. Adapted from Commonwealth of Australia (2017)

The most recent review and feasibility study of the National Food Strategy (FIAL, 2021) focused on food waste avoidance and reuse, and to a lesser extent on the recycling of food waste as animal feed. The approaches established in the review are mainly focused on high-level policies and industry-lead strategies with projections for the reduction of food waste. One of the conclusions of the review is that no single approach on its own, industry or policy-led, will be sufficient to halve food waste in Australia by 2030 and a combination of strategies is essential.

At the household level, policies have shifted to food organics/green organics (FOGO) recycling. Though a recent study has shown that Australia has good to excellent capacities to handle organic food waste (AORA, 2021), several issues in organics recycling have been identified, including physical contamination and impurities in FOGO, local governments do not enforce source separation, and significant access to organic recycling systems between urban and rural areas (Wilkinson et al., 2021). In addition, only 50% of households have access to green bins and only 28% are FOGO (DAWE, 2020), with varying plans across states to make FOGO bins available to all councils.

2.2 How much food waste is produced?

Annually, food waste per capita for selected countries is shown in Figure 2a (UNEP, 2021). Among them, Australia has the largest amount of food waste generation per capita. Globally, each year approximately 46% of produced fruits, vegetables, roots, and tubers go to waste, as well as 35% of fish and seafood, and 30% of cereals (Caldeira et al., 2019; Seberini, 2020). According to data from the Food and Agriculture Administration of the United Nations, production of such amounts of food accounts for 28% of the world's agricultural areas (Aparajita et al., 2023), impacting environments and the balance of the ecosystems as well as humans (with a large quantity of the global population under malnutrition).

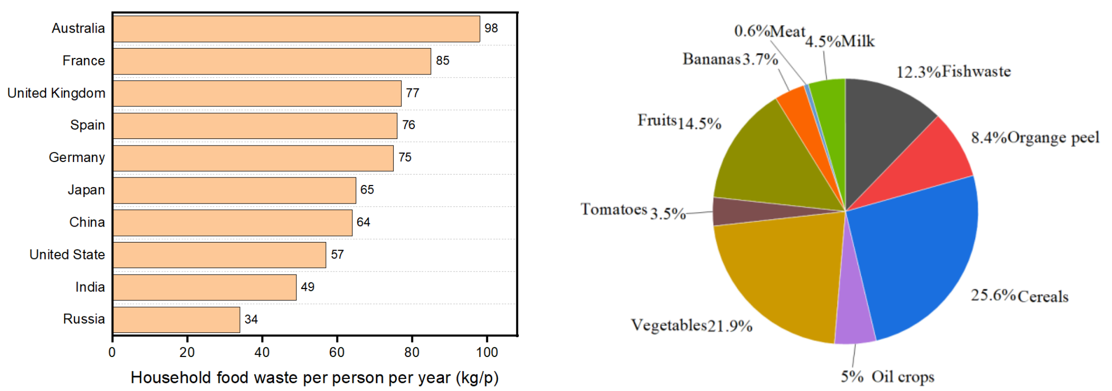

Figure 2. a) Annual generated food waste per capita in several countries, and b) general categories of sources of food waste in 2021

In Australia, 7.6 Mt of food waste are produced every year across the food supply chain and it is estimated that this costs Australian economy around \$36.6 AUD billion each year (FIAL, 2021). Recent estimates from the review of the National Food Strategy suggest 312 kg of food wasted per capita are produced in Australia every year (3 times higher than estimates from UNEP (2021)), costing Australian households $2,000-$2,500 AUD per year. From households alone, 2.46 Mt of food waste are produced annually; with 1.8 Mt of this food waste ending up in landfills (FIAL, 2021).

As a point of comparison, the United States Environmental Protection Agency (US EPA) estimates that approximately 66 Mt of food waste were generated in the food retail, food service, and residential sectors, with about 60% of this waste (around 39.6 Mt) being sent to landfills (US EPA, 2023). In 2021, United Kingdom's biodegradable municipal waste sent to landfill was 6.8 Mt (UK DEFRA, 2023).

In Australia, currently only 28% of local councils separately collect household FOGO. In 2019, a target for providing FOGO to all households and business in Australia was set for 2023 as part of the National Waste Policy Action Plan (ALGA, 2019); however this target has now been shifted to 2030 (ALGA, 2022). Currently, most food waste goes into mixed waste (red bins) and it is bound for landfills. While it is possible to extract organic material from the mixed waste, the mixed waste is not closely monitored and can contain batteries and electrical goods that are sources to heavy metals and will contaminate the organic matter retrieved from the mixed waste (Clarke, 2022). The Australian federal government has also recently released a $67 million AUD Food Waste for Healthy Soils Fund to increase Australia's organic waste recycling rate from 49% to 80% by 2030 through building FOGO infrastructure and fostering a regenerative approach to food systems that return the nutrients embedded in recycled organics to our soils (WMRR, 2021). While there is optimism that FOGO collection targets across Australia will be met by 2030 and that collection cost could be covered by recycling of organic material as compost (Clarke, 2022), outcomes will be subject to the effectiveness of the different state strategies. Table 1 shows the information related to strategies in New South Wales, Queensland, South Australia, and Victoria. Most of the food waste collected is currently composted with all states having official guidelines for design and operation of composting facilities; however, alternatives such as energy recovery from organics using anaerobic digestion are mentioned in most state strategies with varying levels of commitments for the development of new energy recovery plants.

Table 1. State strategies showing targets and current status

State	Strategy	Targets	Status
NSW	NSW Waste and Sustainable Materials Strategy 2041 (NSW DPIE, 2021)	Organic collection in all councils that by offer kerbside collection by 2030	49 councils collect garden organics 39 councils collect FOGO 43 councils do not have organic collection
QLD	Queensland Organics Action Plan 2022 – 2032 (DES, 2022)	80% of households with organic capture services by 2030 50% capture of food waste diversion by 2030	12 councils have green bin collection FOGO being trialled in selected councils
SA	South Australia's Waste Strategy 2020-2025 (Green Industries SA, 2020)	70% landfill diversion in metro Adelaide by 2025	Currently 51% metro Adelaide areas have landfill diversions in place
VIC	Recycling Victoria: A new economy policy (ELWP, 2020)	All households access to organic waste service by 2030	48 councils collect FOGO collection 15 councils collect garden organics 16 councils do not offer green bin collection

Despite advances in the processing of organics and food waste across different states, most of the household food waste generated in Australia continues reaching landfills (FIAL, 2021); therefore, there is need of alternative options for diverting food waste from landfills.

3 USE OF HOUSEHOLD FOOD WASTE DISPOSERS

The use of FWD for diverting food waste can have an important role in circular economy goals and complement actions taken as part of the food waste reduction strategies. The installation rates of FWD vary by country and it has never been fully addressed worldwide. In 2008, it was estimated that 50% of households in the United States were equipped with these units, followed by 30% of households in New Zealand, 12% in Australia, and 10% in Canada (Figure 3) (Carey et al., 2008). Unsurprisingly, the majority of FWD units are installed in urban and metropolitan areas. For example, though Canada shows only a 10% national penetration in the use of FWDs, a study completed in the metro Vancouver area revealed 45% of household dwellings have FWDs and 44% of these dwellings use it on a daily basis (CWWA, 2019). In the United Kingdom, the installation rate of FWDs was reported to be 5% in 2008 (Iacovidou, Ohandja, Gronow, et al., 2012; WRc, 2010), which is considered to be one of the highest in Europe (Carey et al., 2008).

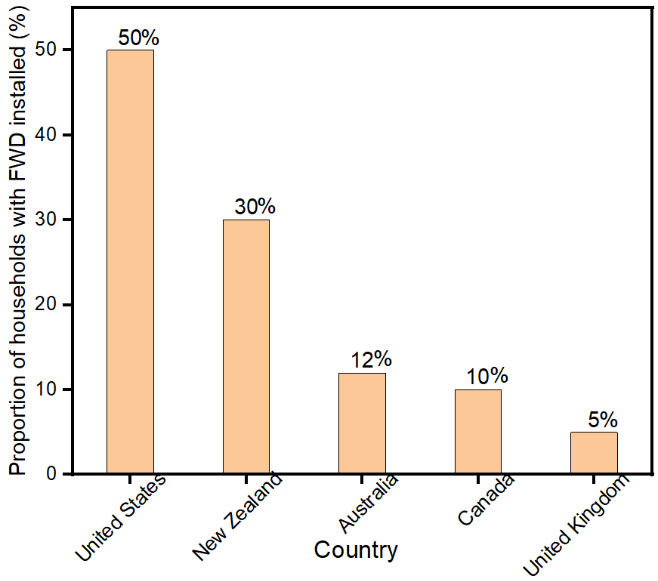

Figure 3. Proportion of households with FWD installed in 2008.

In England, the new Environment Act 2021 (UK Legislation, 2021) states that food waste is to be treated as recyclable waste and must be collected for recycling or composting, which means that rules regulating the use of FWDs may change in the next few year and possibly follow a ban as has happened in Scotland, Wales and Ireland. In the US, regulations for FWDs vary by state and municipality with no strong regulations in many states, some cities completely banning their use and some other states simply discouraging their use. For example, New York City lifted its ban on FWDs use in 1997 for house dwellings (NYC DEP, 1997).

Similarly, the city of Raleigh, North Carolina attempted to ban FWDs in 2008, but rescinded the ban one month later.

In Australia, the installation of FWDs on any facility discharging to the wastewater system in Western Australia is prohibited (Western Australian Legislation, 2013). The Western Australia Water Corporation states that "*this is because food waste disposal units significantly increase the load on their wastewater treatment plants*", therefore "*food waste is better disposed of via the solid waste disposal system, or by home composting*" (Water Corporation, 2024). In other states, regulations seem to depend on the water company or councils. For example, Sydney Water does not allow commercial connections of FWD to their sewer system (Sydney Water, 2023). Brisbane City Council, on the other hand is allowing households to claim a tax deduction through their compost rebate program with the installation of a food dehydrator, composting equipment or FWD (Brisbane City Council, 2024). Nonetheless, as previously discussed in this study, the disposal of food waste in Australia is majorly being encouraged through the implementation of organic curbside collection and centralized processing via composting, incineration, and energy recovery. The implementation of FWDs to divert food waste from landfills is a highly discussed topic. Here some of the most common considerations regarding their use are presented.

3.1 Water consumption

The water consumption per capita associated with FWDs will vary depending on the models, the number of uses per day, the food preparation habits, and number of people in the household. Estimates of the water use per capita from FWDs in Canada range from 2.5 to 15 L/person/day (CWWA, 2019). Evans et al. (2010) summarized a series of studies in the US and Sweden in which FWD water usage was defined as 0.29 L/person/day for big families and 6.4 L/person/day for small families. Similarly, a review of different studies by Iacovidou, Ohandja, Gronow, et al. (2012) showed a water consumption of 1 to 7 L/person/day. In the Australian context a study from the Waverly Council in Sydney (Wainberg et al., 2000) showed that conservatively, the water use per household is 6.2 L/house/day equivalent to 2.95 L/person/day. Later, Lundie and Peters (2005) defined the water consumption from FWD in the same region as 12.4 L/kg of food.

Regarding the overall change in water consumption, some authors have reported no measurable change, some have reported a small increase and some authors have reported a decrease in water use per capita over time. Evans et al. (2010) commented on two studies in

Sweden where the usage per capita was 13 L/person/day lower 11 months after the implementation of FWDs in a community of 100 apartments, yet there was no clear relationship between the decrease and the implementation of FWDs. Similarly, DeOreo et al. (2011) reported that households with FWDs across 10 water agencies in California consumed 49.5 L/house/day than households without FWDs, which is counter-intuitive. In the Californian study, the authors suggested that the presence of FWDs reduced the amount of water run into the kitchen sink to clean out food particles; however, there was not a compelling linkage in the findings.

3.2 Electricity use

For household use, Evans et al. (2010) estimated that the average annual energy consumption from FWDs was 2-3 kWh/household considering FWDs equipped with a 350-500 W motor and an average usage of 2.4 times per day for 16 s on each use. The number of usages is independent of the number of people in the household, as the usage is more related to food preparation. In a similar review, Iacovidou, Ohandja, Gronow, et al. (2012) reported annual electricity values from the literature in the range 2 – 8.5 kWh. For example, Bolzonella et al. (2003) used values of 4.3 – 8.5 kWh to establish the cost of electricity. In an Australian study, Lundie and Peters (2005) reported the electricity consumption of FWDs as 0.02 kWh/kg of food disposed.

3.3 Cost of FWDs

The cost for installing a FWD unit depends on the cost of the device and labour. Prices for InSinkErator brand devices range from $350 to $860 AUD (www.bunnings.com.au). A similar price range can be found from Green Sink Solutions ($330 – $950 AUD) (www.greensinksolutions.com.au). Installation cost is highly variable as it depends on labour and other costs associated with retrofitting a kitchen or additional piping. The recommended hourly price for a plumbing contractor is as a minimum $130 to $140 AUD (PlumbingCosts, 2024). While the replacement of a unit may take approximately one hour, installation will take longer. Considering 5 hours for installation, the cost for installing an FWD could be between $650 and $700 AUD as a minimum. Most plumbers in Australia will also charge a call-out fee between $75 and $95 AUD (PlumbingCosts, 2024). Then, for both the cost of the device and the installation, customers can expect the price to range between $1,055 and $1,745 AUD.

3.4 Sewers

One of the common issues suggested in the literature related to the use of FWD is the increase in sewer pipe blockages. Some studies suggest that the additional food waste content can lead to fat, oil and grease (FOG) blockages (Iacovidou, Ohandja, Gronow, et al., 2012) and significantly more in systems where blockages are a common problem. FOG can accumulate in pumps and cause them to overheat (Evans et al., 2010). Yet, despite these concerns, field studies have not found increases in FOG due to the use of FWDs. For example, the use of camera footage in sewers systems in Shropshire (UK) did not find evidence of FOG blockages as the result of food waste (LGA, 2015). It has been proposed that the implementation of FWD with cold water causes grease and fat to coalesce onto other particles which reduces the chances of attaching to sewer walls (Battistoni et al., 2007; Evans et al., 2010). In terms of suspended solids deposition, incidents of sewer blockages due to increments in suspended solids have not been detected or expected (Marashlian & El-Fadel, 2005). Other investigations using video and photo evidence have not shown significant deposition of solids in pipes (Battistoni et al., 2007; Evans et al., 2010). On the other hand, Bolzonella et al. (2003) showed that some deposition would be expected, depending on the settling velocities and particles sizes. Lastly, another concern from the use of FWD in sewers is the generation of hydrogen sulphide from food waste decomposition along the pipes. This is expected to cause odour and corrosion in clay or metal pipes (Iacovidou, Ohandja, Gronow, et al., 2012). The study from Waverly Council in Sydney concluded that the use of FWDs will result in additional hydrogen sulphide generation, but it was not possible to fully quantify these effects (Wainberg et al., 2000). Impacts on sewer systems from the use of FWD in sewers are possible; however, field evidence suggests that most concerns are unfounded (Evans, 2012).

3.5 Loads on WWTP

The increases in the composition of wastewater characteristics due to the use of FWDs is widely known in the literature, but the impact these changes have in the WWTP vary depending on the scale of the sewer network system and the WWTP. In a typical wastewater treatment process, when wastewater reaches the plant, large objects like rags, plastic and rubbish are removed by using specially designed filter screens. The rubbish and sediment collected are disposed at approved landfill sites. In primary treatment, particles gradually sink to the bottom of the tank and form sludge. The sludge collected in the sedimentation tanks is pumped into huge tanks called digesters. The tanks are heated to grow bacteria, which breaks down the sludge into water and biosolids. Methane gas is a primary by-product of the anaerobic digestion process. The gas is recycled as fuel for heating and mixing inside the digesters or to produce electricity. In other plants without this process, the sludge can be used as fertilizer or ends up in the landfill. In secondary treatment, the wastewater goes through an aeration process by using microbes to remove organic matter. Finally, the water can proceed to advanced treatment or returned to streams and ocean.

In general, increases in total suspended solids (TSS), biochemical oxygen demand (BOD), chemical oxygen demand (COD), nitrogen (N) and phosphorus (P) have been examined. In a review study, Iacovidou, Ohandja, Gronow, et al. (2012) summarized the increases in loading, and the key results are presented in Table 2. The increase of these parameters does not represent a change in the wastewater treatment process; however, significantly more aeration and energy may be necessary during treatment and nutrient removal (Battistoni et al., 2007; Bolzonella et al., 2003; Marashlian & El-Fadel, 2005). Wainberg et al. (2000) concluded that up to a market penetration of 15% household with FWDs, no issues would be expected in the sewage treatment process. A study in Italy suggests that increases in the organic content might be beneficial to nutrient removal and reducing the cost of chemicals use during the wastewater treatment process (Battistoni et al., 2007). In some of the studies conducted in Sweden, BOD, COD, and N loadings at the WWTP did not show increases after the introduction of FWDs and it has been argued that biofilms on sewer walls can adapt to increased loadings, thus producing anaerobic nitrification and denitrification during along the pipelines (Evans, 2012). In terms of additional electricity required to treat loadings at the WWTP, a recent case study estimated an increase in 58% more electricity usage (Kim & Phae, 2023); however, this study used an extreme scenario where all food waste generated from all households was treated by the WWTP. A recent study in Hong Kong, estimated that 0.69-0.9 kWh/kg of food waste was necessary to operate a system combining FWD and WWTP and approximately 64% of the energy was used for aeration (Guan et al., 2022). Similarly, in Australia, a recent study estimated the required electricity to be 0.98 kWh/kg of food waste.

Table 2. Increases in loadings of wastewater composition resulting from the use of FWD.

Wastewater parameters	Increase in loading (%)
TSS	2 - 60
BOD	7.5 – 62
COD	15 – 44
N	1.4 – 19
P	1.2 - 14

Note: adapted from Iacovidou, Ohandja, Gronow, et al. (2012), based on several studies.

3.6 Potential GHG emissions and biogas production gains

It has been recognized in the literature that the increased loadings and concentrations in the wastewater as a result of introducing FWDs will lead to increases in GHG emissions; however, most of the existing studies did not address such increases. In this regard, Kim and Phae (2023) estimated increased emissions of up to 58% when all food waste from households was diverted to the WWTP; however, energy offsets resulting from exploiting additional biogas production were not considered.

Biogas is produced from the anaerobic digestion of organic material during the process of decomposition, which may take place in a special anaerobic digestion plant, landfills, or as part of the wastewater treatment. Benefits of biogas come from exploiting this source as an energy source for heating, cooking or powering turbines in electricity production. Modern technologies and circular economy strategies often propose biogas as part of the synergic process. In Australia, 242 biogas plants were reported in 2017, with a total of 129 located in landfills and half of the biogas produced being flared because of the poor landfill gas quality (Carlu et al., 2019). In contrast, there were 52 WWTPs reported to produce biogas, where 19% is flared and the rest is used for electricity, heating and combined heat and power production (CHP) (Gustafsson et al., 2022). Two examples of WWTPs that can provide energy recovery are the Melbourne Water's Western Treatment Plant, which can generate 72 GWh of electricity from biogas every year to meet about 80% of the electricity needs of the plant, and the Eastern Plant which uses biogas to power about 30% of its electricity usage (Melbourne Water, 2024a). In addition, the Colac Renewable Organics Network (Colac RON) at the Colac Water Reclamation Plant in Victoria is a project set to produce 5.5 GWh of energy annually (Evans et al., 2010). Also, eight of Sydney Water's WWTPs and recycling plants feature cogeneration facilities that transform methane into electricity, boasting a combined capacity of 9.9 MW (NSW CEA, 2024). The other plants capable of processing biogas, are in sector related to industry (dairy, sugar, meat, pulp, paper and beverage), agriculture (animal byproducts and crop residue), and biowaste (households, small scale commercial and industrial activities) (Carlu et al., 2019).

The diversion of food waste to WWTP using FWDs and the net impacts on GHG then need to consider potential energy offsets from the production and use of biogas. A carbon footprint study in Hong Kong, comparing multiple scenarios, revealed the operational energy cost of WWTPs with food waste treatment could increase 1.6 to 2.1 times the cost without food waste; however, the overall GHG emissions could be reduced by 35.2% in comparison to a direct landfill option (Guan et al., 2022). In their study, the option with less GHG emissions was a separate collection system of household food waste and treatment in a centralized anaerobic digestion plant. No composting scenarios were included in the study by Guan et al. (2022).

In Australia, three studies have considered multiple food waste treatment scenarios, showing a diverse range of results. The life cycle assessment by Lundie and Peters (2005) considered the use of FWDs, home composting, centralized composting and landfill, concluding that the use of FWDs was the second-best option in terms of energy usage and GHG emissions. The study by Lundie and Peters (2005), however, did not consider energy offsets at the WWTP. Edwards et al. (2018) also conducted a life cycle assessment of seven different scenarios, including the implementation of FWDs. In general, food waste digestion systems proved to be more efficient in avoiding GHG emissions than composting options. Lastly, Sanciolo et al. (2022) compared GHG emissions from a series of scenarios analysing commercial food waste including landfill, centralized composting and the use of FWDs. They estimated that landfill disposal global warming potential was significantly higher than any other scenario; however, centralized composting was 50% less than the FWDs option.

The analysis of the literature in this report has shown that the implementation of FWDs as a diversion method has significant potential to reduce the food waste ending up in landfills and contribute to circular economies. More importantly, it is clear that no single option for handling food waste is the best or likely to suit all consumers. As such, from the point of establishing strategies for reducing food waste, the consumers should be given options they are willing to use (Evans, 2012).

4 CASE STUDY: SCENARIO COMPARISON

In this case study we provide a calculation of the water and energy cost for a three-person household in Melbourne and the comparison of two food waste management scenarios using landfill and FWDs as the mechanism to divert food waste from the landfill. The calculation of energy consumption and recovery in the analysed scenarios follow the methodology established by Sanciolo et al. (2022). The case study focuses on Melbourne, with the use of information from literature.

4.1 Operation costs for the consumer

In Melbourne, the annual amount of food waste per capita is 207 kg/person (Sheridan et al., 2016) including households and restaurants. Assuming the national ratio of food waste from food services and households is maintained at the city level (FIAL, 2021), 67% of the food waste per capita originates in households. Therefore, we adopt 138.7 kg/person per year as the amount of food waste produced by a person per year in Melbourne.

According to Lundie and Peters (2005) study, the water consumption from the use of FWDs is 12.4 L/kg of food, while the energy consumption is 0.02 KWh/kg of food. The average residential water prices in Melbourne's metro area are $6.77 AUD/kL based on an annual water consumption of 150 kL (ESC, 2024a). For a three-person family the water cost for operating a FWD for a year is $34.9 AUD per year. As for the cost of electricity, considering only the electricity usage in Melbourne of $0.34AUD/kWh (ESC, 2024b), for a three-person household, the annual cost of operation of a FWD is $2.8 AUD per year. Therefore, the total additional cost of operating a FWD for a three-person household is $37.7 AUD per year.

In Table 3, the total operational cost for different penetration costs have been calculated for the city of greater Melbourne Considering the water and electricity cost of operation and FWD penetrations from 5% to 25% of the population in Melbourne, the results indicate potential returns for utility suppliers from $3.3 to $16.5 million AUD/year.

Table 3. Operational cost for consumers at different FWD penetration scenarios.

Household penetration	Water cost (AUD$/year)	Electricity (AUD$/year)	Total (AUD$/year)
5%	$3,056,221	$247,562	$3,303,783
10%	$6,112,443	$495,123	$6,607,566
25%	$15,281,107	$1,237,808	$16,518,915

Note: assuming a population in Melbourne of 5.25 million people.

4.2 Scenario 1. Food waste curbside collection and landfill disposal

In the landfill scenario (Scenario 1), the consumers pay for the weekly collection of a waste bin that is sent to the landfill for waste disposal. Using Melbourne as an example, citywide the curbside waste is taken to a transfer station facility located West Melbourne (City of Melbourne, 2024). This facility is 5.5 kilometres away from the city centre. Additionally, landfill servicing both residential and commercial customers throughout the Melbourne metro area is situated roughly 27.2 kilometres from the transfer station. The energy required for transportation of food waste to the landfill can then be estimated as 0.01 kWh/kg of food waste (calculations in Appendix A). In terms of GHG emissions, following the methods established by Sanciolo et al. (2022), the equivalent carbon dioxide emissions (CO_2e) generated for the transportation of food waste is 0.007 kg CO_2e/kg of food waste (calculations in Appendix A). In addition, methane emissions are equivalent to 2.97 kg CO_2e/kg of food waste from uncontrolled landfills (Moult et al., 2018). The total equivalent CO_2e from landfill disposal is 2.98 kg CO_2e/kg of food waste (calculations in Appendix A).

Just as with the calculation of operation cost to the consumer, we estimate the annual energy necessary for operation and GHG emissions for different percentages of the population (Table 4).

Table 4. Annual energy required for food waste transportation and equivalent CO_2e emissions from landfills.

Household penetration	Food waste (Ton/year)	Required energy for transportation (kWh/year)	Equivalent CO_2 emissions (TonsCO_2e/year)
5%	36,406	364,061	108,490
10%	72,812	728,122	216,980
25%	182,030	1,820,306	542,451

Note: assuming a population in Melbourne of 5.25 million people.

4.3 Scenario 2. Diversion with FWDs and treatment at WWTP

In this FWD scenario (Scenario 2), the food waste is ground with the FWD and transported to the WWTPs via the sewer system where the food waste is treated. There is energy recovery from biogas and electricity production at the plant. WWTPs in Melbourne are equipped with capacity for producing electricity (Melbourne Water, 2024a). Following the methods from Sanciolo et al. (2022), the energy necessary to process food waste and equivalent CO_2 emissions can be estimated in terms of the Chemical Oxygen Demand (COD) of the food waste, i.e. the amount of oxygen necessary to treat the organic material in the reactor of the WWTP. A study by Graunke and Wilkie (2014) establishes that the COD per kilogram of food waste of 0.35 kgCOD/kg of food waste. In addition, a benchmark study of WWTPs in Australia showed that amount of power required for COD removal is 2.8 kWh/kg of COD removed (de Haas et al., 2018). Sanciolo et al. (2022) adapted a value of 1.3 kWh/kg of COD removed as the output energy generation from food waste at the WWTP. Lastly, the Department of Environment, Land, Water and Planning set the GHG co-efficient as 1.06 kgCO_2e/kWh for 2022 (ESC, 2022). As such, the calculations for the net electricity required in the operation of the WWTP and GHG emissions are 0.53 kWh/kg of food waste and 0.56 kgCO_2e/kg of food waste (calculations in Appendix A). The estimates of the annual net energy consumption and equivalent CO_2 emissions from the WWTP are shown in Table 5.

Table 5. Net energy consumption and equivalent CO_2 emissions from the WWTP are shown in Table 5

Household penetration	Food waste (Ton/year)	Net energy consumption (kWh/year)	Equivalent CO_2 emissions (TonCO_2e/year)
5%	36,406	19,295,246	20,387
10%	72,812	38,590,492	40,774
25%	182,030	96,476,231	101,937

Note: assuming a population in Melbourne of 5.25 million people.

4.4 Analysis

In this analysis, estimates for FWD operation cost for consumers were established at different penetration levels. Based on both the prices of water and electricity, the total cost for consumers was calculated for different levels of FWD penetration, therefore, the potential gains for utility companies range between $3.3 to $16.5 million AUD/year depending on FWDs penetration of 5% to 25%. Therefore, there are potential returns for investing in upgrading WWTPs systems and promoting the diversion of food waste from landfills via FWDs. The costs for upgrading and constructing WWTP are highly variable and often dependent on the scale of the system. For comparison, here we estimate the capital cost and operation cost of adapting tertiary treatment including a digester to produce biogas in a WWTP based on the study from Ozgun et al. (2021) (calculations in Appendix). Considering the additional wastewater resulting from FWD operation (using 12.4 L/kg of food waste), the capital cost for upgrading and operating the tertiary treatment is shown in Table 6. These high-level calculations show results show a high feasibility for investing in upgrading WWTPs.

Table 6. Capital cost and cost for operation for a tertiary treatment including digester.

Household penetration	Additional wastewater (m^3/day)	Total cost to consumers (water + electricity) (AUD/year)	Capital cost for tertiary treatment with digester (AUD)	Capital cost of operation tertiary treatment with digester (AUD)
5%	1,237	$3,303,783	$3,407,624	$808,567
10%	2,474	$6,607,566	$5,749,220	$1,144,121
25%	6,184	$16,518,915	$11,478,746	$1,810,340

Note: assuming a population in Melbourne of 5.25 million people.

The comparison of the two scenarios showed a great distinction in terms of energy requirements and equivalent GHG emissions. The energy usage in Scenario 1 is significantly lower than in Scenario 2, even after considering energy recovery from biogas production. One limitation of this study is that the energy required to operate the transfer facility and the landfill is not accounted for, which could increase the energy consumption due to transportation. However, landfill operations are often considered of low energy consumption (0.0033 kWh/kg of waste according to Guan et al. (2022)), which may not have a significant impact in the calculations presented here. On the other hand, in Scenario 2, individual processes were not simulated within the WWTP and it is considered that all food waste is treated in the reactor, while in reality some of the food waste is removed in primary treatment as part of the sludge and treated as biosolids, yet Evans (2012) argues that almost all food waste is converted to biogas as the solid part of the food waste is highly biodegradable. In addition, newer technologies such as the use of anaerobic membrane treatment have shown that the inclusion of food waste in the wastewater process can increase biodegradability (Moñino et al., 2017), making the process more efficient which will require less energy. However, this is outside the scope of this study. The results show that FWDs penetration of 10% is feasible in Melbourne as the maximum energy that can be produced in the West Treatment Plant is 70 million kWh

(Melbourne Water, 2024a). However, Melbourne Water has recently announced a $711 million AUD investment for increasing the capacity of the Western Waste Water Treatment Plant by 50% which will be able to process 300 billion litres per year (Melbourne Water, 2024b). These upgrades have been implemented as the population of Melbourne is projected to double by 2050. The increases in the amount of wastewater as the result of FWDs up to a penetration of 25% will be less than 1% of the upgraded capacity of the plant.

In Scenario 1, almost five times more equivalent CO_2 emissions are produced than in Scenario 2. This analysis does not include methane recovery in Scenario 1 and emissions that may occur in Scenario 2 during the time food waste travels in the sewer system are neglected. No values have been reported in the literature for emissions in sewers solely as the result of additional food waste, therefore these were not calculated, and they are not expected to be significant.

The analysis of these scenarios shows that the diversion of food waste from landfills via FWDs is feasible in Melbourne up to a realistic 25% penetration. The treatment of food waste at the WWTP after the implementation of FWDs is costly in terms of electricity requirements, but much lower emissions are achieved. Based on the limitations discussed here, it is certain that the optimal solution for food waste diversion should be a combination of strategies rather than a single option. This highlights the arguments made by Evans (2012) that citizens must be provided with choices they are willing to use.

5 DEVICE TESTING

Testing of the devices was conducted at the laboratory facilities of the School of Engineering, Deakin University. The testing methods were adapted from the *Australian Standard 2396 – Performance of Household Waste Disposal Units*. The AS 2396 describes a series of tests for the performance of FWDs. It is noticed that the AS 2396 is currently categorised as a "withdrawn standard" and as such it was used mainly as a guideline for establishing the testing procedure and measurable parameters.

In this project, two different tests were adapted from the AS 2396, the *Degree of fineness test* and the *Grinding rate test,* as these were considered the most relevant in terms of evaluating the performance of the devices for grinding food waste (Table 7). Other tests described in the AS 2396 were not taken into consideration. It is also assumed that the imported devices have been previously tested and meet the *American National Standard ASSE 1008 – 2020 Performance Requirements for Plumbing Aspects of Residential Food Waste Disposer Units*.

The testing arrangement and number of samples were discussed and agreed upon by Deakin University and Abey Australia given the time constraints of the project and required data. In addition to the tests adapted from AS 2396, two different pipe configurations were designed using clear pipes to observe potential issues in food waste clogging the pipes after grinding and before reaching a drain stack or branch. The two clear pipe designs are in accordance with the maximum length for pipes from weir to stack as established in the National Construction Code (ABCB, 2022) and modified according to the AS/NZS 3500.2 (ASNZS, 2021). The clear pipe configurations consist of a 3 m length pipe with two 90° bends at a 2.5% grade and a 6 m length pipe with no bends (Figure 4). The list of tests conducted are shown in Table 7 and details of the quantity and food samples is shown in Table 8.

5.1 Food samples

For the tests adapted from the AS 2396, the food samples were prepared in accordance with the procedure described by the standard (Table 8). In addition, at the request of Abey Australia, an additional type of food sample was prepared using specific food items and the *Grinding Rate test* was performed on those samples. For the clear pipe tests, the food samples varied slightly from the AS 2396 *Degree of Fineness test* samples, where the amounts of food waste were reduced as the clear pipe testing was focused on observing potential food blockages, not the grinding performance.

Table 7. Selected tests from the AS2396.

AS 2396 - 1980		
Test	**Conducted in the project**	**Description**
Degree of Fineness test		Measures the size distribution of ground food.
Test load	Yes	
Grinding Rate test (ground food in a minute)		Grinding is conducted over a minute to calculate the percentage of the sample ground during a minute.
Bone grinding test	Yes	
Citrus rind test	No	
Mixed load test	Yes	
Grinding Rate test (grinding time to completion)		Grinding is conducted to completion and time to fully grind the food is reported.
Bone grinding test	Yes	
Mixed load test [1]	Yes	

Note: [1] The grind to completion test is only described in the standard for rib bone samples. The mixed load was included to test the capabilities of the devices.

Figure 4. Left, 3 m long pipe setup with 2 bends (DN 50). Right, 6 m long straight pipe setup (DN 50).

Table 8. Number of samples per device and scheduled tests.

Sample	Number of samples tested
Grinding rate test (Mixed standard load test)	
150 g banana peel	3 samples per device 1 sample ground to completion 2 samples ground for a minute
100 g corn cobs	
100 g grapefruit rind	
50 stalk celery	
10 g - 2 paper towels and 2 tea bags	
190 raw potatoes	
Grinding rate test (Mixed Abey load test)	
100 g meat scraps	3 samples per device 1 sample ground to completion 2 samples ground for a minute
2 apple cores	
2 eggshells	
100 g melon rinds	
30 g pistachio shells	
2 whole prawns	
Grinding rate test (Bone grinding)	
150 g rib bones	3 samples per device 1 sample ground to completion 2 samples ground for a minute
Degree of fineness (Mixed fineness load)	
125 g rib bones	2 samples per device
125 g celery	
125 g carrot	
125 g lettuce	
Clear pipe testing (3 m pipe with bends)	
50 g rib bones	2 samples per device
75 g celery	
75 g carrot	
75 g lettuce	
25 g pistachio shells	
Clear pipe testing (6 m straight pipe)	
50 g rib bones	2 samples per device
75 g celery	
75 g carrot	
75 g lettuce	
25 g pistachio shells	

Food sample preparation was carried out following the procedures established by the AS 2396, with slight variations for testing the food items requested by Abey Australia. Some changes were also made to the size and weight of some items as well as the number of samples depending on the availability of products.

Beef rib bones were specially ordered so that the length of the bones was between 2.5 cm and 5 cm. Most of the bone pieces were approximately 4 cm long. The bones were prepared by boiling them in water in a covered pot for 1 hour, then replacing the water and boiling them for another hour. After the second boiling, bones were rinsed with hot water, and the remaining gristle was removed. Bones were then dried in a preheated oven at 150 °C for 1 hour before grinding. Meat trimmed from rib bones was boiled for 20 minutes and used as meat scraps.

Grapefruits between 10 to 11.5 cm in diameter were selected and cut in quarters; rinds were separated, with all internal segments removed. The desired mass of each sample was adjusted by adding trims from other rinds. Melon rinds were prepared similarly to grapefruit rinds by cutting the melons in eights and removing the seeds and pulp. Corn cobs were purchased precut in lengths of 12 cm and cut in half. Corn cobs were put in boiling water for 10 minutes and kernels were removed. The total mass was adjusted by trimming one end or adding from another piece. Celery stalks were cut to lengths of 10 cm. Potatoes used were uncooked, between 5 and 6 cm in diameter and cut into quarters. Tea bags were 1.8 g on average, submerged in hot water at 95 ±5 °C for 10 minutes, drained for 5 minutes, and all strings and tags were removed before grinding.

5.2 Testing procedure

The testing procedures for each of the tests are presented below. Mixed loads refer to a combination of food items that are ground using the devices. There is a standard mix load food sample defined by the AS 2396 used for the Grinding Rate test and a mix load defined using the requested food items by Abey Australia. In addition, samples consisting of only rib bones were prepared for the Grinding Rate test. The Grinding Rate test also has two different procedures for grinding the food to completion or grinding the food over a minute.

The Degree of Fineness test uses an additional type of mix load food sample. Two samples were tested for each device.

5.2.1 Grinding Rate test to completion (Standard and Abey mix load)

The grinding chamber was preloaded with part of the test load making sure the food within the chamber was loose and did not jam the device. Water was turned on and measured at a rate of 9.0 L/min (0.15 L/s). The FWD unit was turned on and after 10 seconds the remaining test load was added. If a jam occurred, this was cleared before proceeding. The device was turned off when only a small amount of material remained, and water was run for 1 minute to clear loose material.

The material was then removed, dried on a paper towel and weighed. If the unground material weighed more than 4.5 g, it was returned to the chamber and ground for another 15 seconds. The procedure was repeated until the unground material weighed 4.5 g or less. The total grinding time was then recorded.

5.2.2 Grinding Rate test in 1 minute (Standard and Abey mix load)

The grinding chamber was preloaded with part of the test load (m_L) making sure the food within the chamber was loose and did not jam the device. Water was turned on and measured at a rate of 9.0 L/min (0.15 L/s). The FWD unit was turned on and after 10 seconds the remaining test load was added and ground for an additional 50 s. Water was then run for 1 minute to clear loose material, the material was collected, squeezed dried by hand and weighed (m_c). The mass of ground material (m_g) was determined:

$$m_g = m_L - m_c$$

as well as the percentage of material ground in 1 minute (P_1):

$$P_1 = \frac{m_g}{m_L} \cdot 100$$

The value of P_1 following this procedure should not be less than 95%.

The value of m_g was also used to calculate the amount of food ground in 1 minute.

In the case of the Abey mix load, the test was conducted for 30 seconds as the sample size was only half the mass of the standard mix load.

5.2.3 Grinding Rate test to completion (rib bones)

The device was turned on and water was adjusted to a flow rate of 9.0 L/min (0.15 L/s). The test load was then fed to the device. If a jam occurred, this was cleared before proceeding. The device was turned off when only a small amount of material remained. Water was run for 1 minute to clear loose material.

The unground material was then removed, dried on a paper towel and weighed. If the unground material weighed more than 4.5 g, it was returned to the chamber and ground for another 15 s. The procedure was repeated until the unground material weighed 4.5 g or less. The total grinding time was then recorded.

5.2.4 Grinding Rate test in 1 minute (rib bones)

The device was turned on and water was adjusted to a flow rate of 9 L/min (0.15 L/s). The test load was fed to the device as quickly as possible and ground for 1 minute. If a jam occurred, it was cleared before proceeding. Water was then run for 1 minute to clear loose material.

The unground material was collected and dried in a preheated oven at 150°C for one hour and weighed (m_c). The mass of ground material (m_g) was determined:

$$m_g = m_L - m_c$$

as well as the percentage of material ground in 1 minute (P_1):

$$P_1 = \frac{m_g}{m_L} \cdot 100$$

The value of P_1 following this procedure should not be less than 65%.

The value of m_g was used as the amount of bones ground in 1 minute.

5.2.5 Degree of Fineness test

First, the clean, dry mass of four test sieves was determined. The sieves required apertures of 13.2 mm, 6.7 mm, 2.36 mm and 425 µm and stacked in descending order.

The grinding chamber was preloaded with part of the test load making sure the food within the chamber was loose and did not jam the device. Water was turned on and measured at a rate of 9.0 L/min (0.15 L/s). The device was turned on and the rest of the food sample was added. Food waste was collected and sieved. The device was stopped when the total grinding time was 3 minutes.

After the material was collected the food waste was washed through the sieves using a spray nozzle. After washing, each sieve was drained on a horizontal surface for 10 minutes and weighed wet to within ± 0.5 g.

The total mass of waste retained (m_t) was then determined by considering the summation of the mass of sieves plus wet waste minus the mass of the dry clean sieves. Then, the net mass of wet waste material in each sieve was calculated.

The amount of waste retained on the sieves shall not exceed the values given Table 9.

Table 9. Maximum food waste retained on sieves

Sieve aperture size (mm)	Maximum waste retained (%)
13.2	10 (retained by one sieve)
13.2 + 6.7	25 (retained by the two sieves)
13.2 + 6.7 + 2.36	75 (retained by the three sieves)

5.2.6 Clear pipes test

The grinding chamber was preloaded with part of the test load making sure the food within the chamber was loose and did not jam the device. Water was turned on and measured at a rate of 9.0 L/min (0.15 L/s). The unit was turned on and after 10 seconds the remaining test load was added and ground for an additional 50 s.

After food grinding, the pipes were inspected and any material remaining along the pipes was identified. If ground material remained on the pipes, the water was turned on for 15 seconds to flush the retained material. The number of flushes necessary to completely clear the pipes was recorded.

5.3 Results and Discussion

Results and key observations from the different tests are presented in the following sections.

5.3.1 Grinding rate test for mixed load food samples

A total of 12 mix load samples were prepared as described in section 5.1. An example of the standard mix load and Abey mix load food sample is shown in Figure 5. Six standard mix and six Abey mix load samples were prepared, three samples to test each FWD device. The mass for each of the food items in the mixed loads is presented in Appendix B.

Figure 5. Left: Standard load food sample (banana peels, grapefruit rinds, celery stalks, raw potato, corn cobs, two tea bags and two paper towels). Right: Abey load food sample (prawns, egg shells, melon rinds, apple cores, meat scraps, pistachio shells).

For the WDU-3ASAU, ground-to-completion of the standard mix load samples took 2 minutes and 15 seconds, leaving 0.68 g of corn cobs at the end of the test. For the WDU-4ASAU, the grind to completion time was 1 minute and 30 seconds, with a residual weight of 0.73 g at the end of the test.

In terms of the percentage of ground material during a minute, neither device met the desired 95% of the standard mix load as required by the AS 2396. The WDU-4ASAU device showed a better performance and had the best grinding rate of 89% while the WDU-3ASAU showed the poorest performance of 64% (Table 10). The WDU-3ASAU device presented some issues grinding the large standard mix load samples, especially the corn cob material. This delayed the loading of the sample into the machine. Under normal operation, food waste quantities may not be as large therefore the delay in feeding the grinder may not be an issue. Pieces of grapefruit rind were also observed to be part of the unground food waste collected from both device chambers at the end of the tests. The fibrous nature of the grapefruit rinds delayed the grinding time in both devices. The grinding rates obtained from the standard mix load samples show an average of 425.56 g/minute for the WDU-3ASAU and 517.45 g/minute for the WDU-4ASAU.

Table 10. Grinding rate test results from standard load food sample

Device / Samples	WDU-3ASAU			WDU-4ASAU		
	1	2	3	4	5	6
Time to grind to completion	2 min + 15 s			1 min + 30 s		
Collected material (m_c) (g)		219.21	135.62		98.86	68.66
Ground material (m_g) (g)		385.02	466.09		502.80	532.10
Percentage material ground in 1 minute (P_1) [1]		64%	77%		84%	89%
Grinding Rate (grams/minute)		385.02	466.09		502.8	532.1

Table 11. Grinding rate test results from Abey load food sample

Device / Samples	WDU-3ASAU			WDU-4ASAU		
	1	**2**	**3**	**4**	**5**	**6**
Time to grind to completion	2 min + 30 s			3 min stopped		
Collected material (m_c) (g)		52.74	36.10		36.57	43.65
Ground material (m_g) (g)		273.43	258.30		281.99	246.30
Percentage material ground in 1 minute (P_1) [1]		84%	88%		87%	86%
Grinding Rate [2] (grams/minute)		546.86	516.60		506.76	520.32

Note: [1] Abey mix food sampled are smaller than the standard samples (Appendix B), therefore the time of the test was adjusted to 30 s to proportionally to resemble the standard method. [2] The grinding rate grams/minute was extrapolated from the duration of the test.

While testing the devices using the Abey mix load sample, the fibrous material of the melon rinds was the food item that took the longest to grind. The WDU-3ASAU took 2 minutes and 30 s to completely grind the sample, after which the remaining unground material only amounted to 3.52 g. The grind-to-completion test on the WDU-4ASAU was stopped after 3 minutes because after multiple 15 s bursts, the weight of the material in the chamber did not decrease. The final weight of unground material was 10.19 g and mainly consisted of small melon rind pieces (Figure 6). It was observed that melon rinds tend to accumulate on the sides of the device, delaying the grinding (Figure 6). Once melon rinds are reduced to a small size with smooth edges and the rest of the food is ground, the pieces of food bounce within the chamber and the grinding becomes less effective.

Figure 6. Left: Residual unground food waste after 3 minutes of grinding in the WDU-4ASAU device. Right: Melon rinds being pushed to the walls of the device chamber.

The WDU-3ASAU unit showed a much better performance in grinding the Abey mix load than the standard mix load food sample test and similar results to the WDU-4ASAU, yet both devices were below 95% of ground material at the end of the test (Table 10 and 11). Noticeably, as the Abey mix load sample is smaller, nearly half of the standard load food sample, the grinding time was adjusted to 30 seconds to maintain similar testing conditions compared to the standard test and the grinding rate over one minute was then estimated proportionally to the time. Both machines showed a very similar performance in grinding the Abey mix load food sample. The average grinding rate for the WDU-3ASAU and WDU-4ASASU is 531.73 g/minute and 513.54 g/minute respectively.

Given the measured grinding rates from both mix loads and motor denomination in the devices, the energy required to grind a kilogram of food waste can be estimated as 0.030 kWh/kg of food for the WDU-4ASAU and 0.019 kWh/kg of food for the WDU-3ASAU, which is within the range from previous studies (Lundie & Peters, 2005).

5.3.2 Grinding rate for rib bones

For the samples only containing rib bones (Figure 7), both devices meet the requirements set by the standard, grinding above 65% of the bone material in 1 minute and ground to completion in less than 5 minutes. The WDU-4ASAU device can grind 74% of the standard sample in one minute on average, while the WDU-3ASAU device was able to grind 71% of the standard sample in one minute on average (Table 12). While the results of grinding over a minute showed very similar outcomes in both devices, the WDU-4ASAU device was able to grind the sample completely in only 2 minutes and 15 seconds, a minute faster than the WDU-3ASAU device. The average rib bone grinding rate for the WDU-3ASAU and WDU-4ASASU are 90.85 g/minute and 111.89 g/minute respectively.

Figure 7. Left: Example of rib bones load food samples for bone grinding rate test. Right: Mixed load food sample for the degree of fineness test.

Table 12. Grinding rate test results from rib bones sample

Results	**WDU-3ASAU**			**WDU-4ASAU**		
	1	2	3 [1]	4	5	6
Mass of load (m_L)	151.45	151.41	103.94	150.21	149.55	151.45
Time to grind to completion	3 min + 15 s			2 min + 15 s		
Collected material (m_c) (g)		49.06	52.89		44.59	32.73
Ground material (m_g) (g)		102.35	51.05		104.96	118.81
Percentage material ground in 1 minute (P_1) [1]		68%	74%		70%	78%
Grinding Rate (grams/minute) [1]		102.35	79.34		104.96	118.81

Note: [1] The time of the test was adjusted proportionally for Sample 3 to 40 seconds to resemble the standard method.

Similarly to the analysis of mixed loads, the required energy to grind a kilogram of bones was calculated as 0.10 kWh/kg of food waste for the WDU-3ASAU device and 0.14 kWh/kg of food waste for the WDU-4ASAU device. This translates into 5 times the energy cost for grinding food waste not containing bones.

5.3.3 Degree of fineness

Mix load food samples for the Degree of Fineness were prepared (Appendix B). An example of the mix food sample is presented in Figure 7. Two samples were tested using each FWD device. The sieve ensemble for filtering the solids was used as established by the AS 2396, with decreasing sieve aperture (13.2 mm, 6.7 mm, 2.36 mm and 425 µm).

The results for each of the samples are presented in Tables 13 to 16. Almost no material remained in the 13.2 mm sieves, but some remaining bone pieces were left in the chamber after 3 minutes of grinding. The devices meet most of the conditions established by the standard, except in Samples 2 and 4 where the combined material in sieves 13.2 mm, 6.7 mm and 2.36 mm was slightly higher than 75%. Between 56% and 73% of the ground food waste material was retained by the 2.36 mm sieve in all tests. The main reason for this high quantity retained is that fat and oils from the rib bones congeal and trap particles that could pass the sieve aperture into particle groups. These groups of accumulated particles are not easily rinsed with room-temperature water. The fat and oils also delay the water draining from the sieve, therefore the perceived weight capture in the 2.36 mm sieve can be due to excess water. Overall, the performance of the two devices met the standard in at least one of the samples tested.

Figure 8. Left: Resulting food waste in sieves for Sample 2 (WDU-3ASAU). Right: Resulting food waste in sieves for Sample 3 (WDU-4ASAU). Note: top left sieve 13.2 mm, top right 6.7 mm, bottom left 2.36 mm and bottom right 425 µm.

Table 13. Degree of fineness Sample 1 (WDU-3ASAU)

Sieves	Initial weight (g)	Final weight (g)	Net wet mass retained (g)	% retained
13.2 mm	654.18	663.38	9.20	1.49
6.7 mm	559.20	582.52	23.32	3.78
2.36 mm	520.84	868.64	347.80	56.43
425 µm	482.56	718.56	236.00	38.29
Total	2216.78	2833.10		
Total mass of wet weight (m_t) = 616.32 g				
Percentage retained (13.2mm)			1.5 %	< 10%
Percentage retained (13.2mm + 6.7mm)			5.3 %	< 25%
Percentage retained (13.2mm + 6.7mm + 2.36mm)			61.7 %	< 75%

Note: After 3 minutes 2.64 g of unground material remained in the chamber

Table 14. Degree of fineness Sample 2 (WDU-3ASAU)

Sieves	Initial weight (g)	Final weight (g)	Net wet mass retained (g)	% retained
13.2 mm	656.34	671.83	15.49	2.38
6.7 mm	561.43	587.66	26.23	4.03
2.36 mm	520.6	996.03	475.43	72.97
425 µm	489.21	623.61	134.4	20.63
Total	2227.58	2879.13		
Total mass of wet weight (m_t) = 651.55 g				
Percentage retained (13.2mm)			2.4 %	< 10%
Percentage retained (13.2mm + 6.7mm)			6.4 %	< 25%
Percentage retained (13.2mm + 6.7mm + 2.36mm)			79.4 %	> 75%

Note: After 3 minutes 5.87 g of unground material remained in the chamber

The potential effect these results can have on the rest of the process is a gap in the knowledge. It is unlikely that the particle size between 2.36 mm and 6.7 mm would result in issues for most pump systems. On the other hand, if settling velocity is high, most of these particles can be captured by the grit removal process at WWTP, therefore not reaching the primary sedimentation and the energy recovery process. The 2.36 mm particle size is borderline on the larger size of particles that commonly get formed during municipal wastewater treatment (Guo et al., 2020; Smoczynski et al., 2016). It is clear from the observations that most of the material retained in the 2.36 mm sieve has smaller particle sizes, but they group due to the fat and oils. During the process of transportation to the WWTP, it is unknown if these particles will come apart or continue clumping. This is outside of the scope of this study.

Table 15. Degree of fineness Sample 3 (WDU-4ASAU)

Sieves	**Initial weight (g)**	**Final weight (g)**	**Net wet mass retained (g)**	**% retained**
13.2 mm	656.65	671.26	14.61	2.49
6.7 mm	560.53	578.15	17.62	3.00
2.36 mm	521.41	913.78	392.37	66.78
425 µm	485.93	648.88	162.95	27.73
Total	2224.52	2812.07		
Total mass of wet weight (m_t) = 587.55 g				
Percentage retained (13.2mm)			2.5 %	< 10%
Percentage retained (13.2mm + 6.7mm)			5.5 %	< 25%
Percentage retained (13.2mm + 6.7mm + 2.36mm)			72.3 %	< 75%

Note: After 3 minutes 9.15 g of unground material remained in the chamber

Table 16. Degree of fineness Sample 4 (WDU-4ASAU)

Sieves	**Initial weight (g)**	**Final weight (g)**	**Net wet mass retained (g)**	**% retained**
13.2 mm	656.38	667.32	10.94	1.66
6.7 mm	560.15	585.22	25.07	3.80
2.36 mm	520.29	982.51	462.22	70.08
425 µm	485.47	646.81	161.34	24.46
Total	2222.29	2881.86		
Total mass of wet weight (m_t) = 659.57 g				
Percentage retained (13.2mm)			1.66 %	< 10%
Percentage retained (13.2mm + 6.7mm)			5.5 %	< 25%
Percentage retained (13.2mm + 6.7mm + 2.36mm)			75.5 %	> 75%

Note: After 3 minutes 3.92 g of unground material remained in the chamber

5.3.4 Clear pipes test

During the clear pipe test, no food waste residue was observed along the 6 m straight pipe test after grinding. Bone and pistachio shell particles remaining on the S bend were flushed after one flush of 15 seconds of running the water at 9.0 L/minute (0.15 L/s). Collecting water in the sink with the stopper and releasing it also resulted in the complete flushing of any food particles on the S bend.

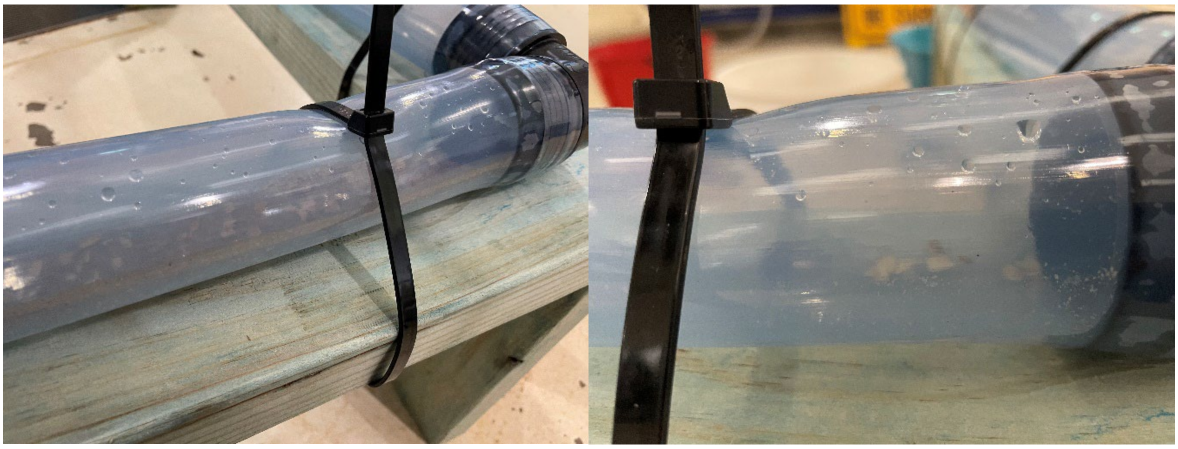

Figure 9. Left: Rib bone and pistachio shell particles remaining on the 90-degree bend after grinding. Right: Rib bone and pistachio shell particles remaining on the 90-degree bend after 5 flushes.

Table 17. Food load sample weight for straight clear pipe

Food weight (g)	**WDU-3ASAU**		**WDU-4ASAU**	
	1	2	3	4
Rib bones	34.48	46.48	31.04	26.44
Celery	76.48	75.15	74.33	76.43
Carrot	75.14	75.41	75.46	75.4
Lettuce	76.56	76.84	76.98	76.08
Pistachio shells	25.05	25.15	25.62	25.14
Total	262.66	273.88	257.81	254.35

Table 18. Food load sample weight for clear pipe with bends

Food weight (g)	**WDU-3ASAU**		**WDU-4ASAU**	
	1	2	3	4
Rib bones	53.99	43.63	54.75	41.83
Celery	75.74	75.88	74.68	74.54
Carrot	75.62	75.06	76.13	75.67
Lettuce	75.14	75.33	75.48	76.13
Pistachio shells	25.35	25.19	25.22	25.06
Total	305.84	295.09	306.26	293.23

In the clear pipe test with bends, accumulation of rib bones and pistachio shell particles were observed at the 90-degree bends (Figure 9). The largest issue, in this case, was the connection type used with the flexible pipes as the 90-degree bend goes into the pipe and creates an obstacle for the food particles. This would not normally happen in a conventional 90-degre connection where the connection covers the pipe. However, after only 5 to 6 flushes only a few pieces of food waste remained close to the bend (Figure 9). No potential issues were observed flushing the food waste material on the S bend.

6 CONCLUSIONS AND RECOMMENDATIONS

6.1 Conclusions

Following the desktop study and the testing of the FWD devices, the following conclusions can be drawn:

FWD Adoption and Implementation:

- Food waste disposers (FWDs) have the potential to divert food waste from landfills and contribute to circular economies.
- Adoption of FWDs varies globally, with high penetration in the US and Canada, moderate in Australia (12% of households), however, banned in Western Australia.
- Local policies on FWDs vary within Australia, with some water utilities such as Sydney Water banning them, while in Brisbane, the city council is promoting them.

Food Waste Management in Australia:

- Australia's strategies focus on reducing food waste at production and retail levels, encouraging food donations, and researching new technologies.
- Households generate 2.46 Mt of food waste annually, with 1.8 Mt going to landfills.
- Implementation of Food Organics/Green Organics bins (FOGO) is inconsistent across states and may not meet the targets for 2030.
- Additional strategies at smaller scales are needed.

Operation of FWDs:

- Results from the literature indicate water usage by FWDs is 12.4 L/kg of food waste, and electricity usage is 0.02 kWh/kg of food waste.
- Annual operational cost of a FWD for a three-person household is $37.7 AUD. Initial purchase and installation costs ranges between $1,055 and $1,745 AUD based on current approximations of unit price and labour cost.
- Concerns about FWD impacts on sewers are largely unfounded.
- Increased loadings in wastewater treatment plants (WWTPs) due to FWDs can increase electricity consumption but can be offset by energy recovery technologies.

Comparative Scenarios for Melbourne:

- Landfill disposal via curbside collection with no biogas recovery requires considerably less energy than FWD usage with biogas recovery at WWTP; however, in the second scenario, there are significantly fewer greenhouse gas emissions as a result of energy recovery.

Performance of FWD Devices:

- Two FWD models were tested, WDU-3ASAU and WDU-4ASAU. Both models achieved good grinding rates for various food waste (approximately 500 g/kg of food waste in average).
- The percentage of food waste ground in a minute was below the percentage required by the standard AS 2396-1980.
- Energy requirements for the use of the WDU-3ASAU and WDU-4ASAU are 0.19 kWh/kg of food waste and 0.03 kWh/kg of food waste. This is close to previously reported values in literature.
- Grinding rates of bones show five times the energy requirement than for other food waste.
- Degree of fineness tests met the standard requirements in at least one of the tested food samples. Most material was collected in the 2.36 mm sieve. The implications of the particle size in the treatment process are unclear.
- In the clear pipes test, some accumulation of particles was observed in pipes. No potential for clogging was observed.

6.2 Recommendations

Recommendations arising from this study are established below:

- Installation of FWDs in households is feasible, particularly in regions where energy recovery systems are in place at WWTPs.
- Implementing rebate programs like Brisbane can help incentivize their adoption.
- Work towards consistent food waste management policies across Australian states should continue the adoption of FOGO bins. However, a single solution may not be feasible nationwide. Providing diverse options that consumers are willing to use can be helpful.
- Conducting further research on the potential effects of food waste sludges resulting from FWD on wastewater treatment and energy recovery processes would help to gain a deeper and broader understanding of the impact.

- Discourage the grinding of bones using the tested devices as this can result in significantly large energy consumption, increasing operation costs. Such materials are better placed in the bin.

References

ABCB. (2022). NCC 2022 Volume Three - Plumbing Code of Australia. In *National Construction Code 2022*. Australian Building Codes Board.

ALGA. (2019). *National Waste Policy Action Plan*. Australian Local Government Association (ALGA). https://www.dcceew.gov.au/sites/default/files/documents/national-waste-policy-action-plan-2019.pdf

ALGA. (2022). *National Waste Policy Action Plan - Annexure*. Australian Local Government Association (ALGA). https://www.dcceew.gov.au/sites/default/files/documents/national-waste-policy-action-plan-annexure-2022.pdf

AORA. (2021). *Australian Organics Recycling Industry Capability Assessment 2020-21 Page*. Australian Organics Recycling Association.

Aparajita, L., Santhanaraj, D., Rajakumar, K., Ramkumar, V., Adinaveen, T., & Leema, S. P. (2023). A critical review on food waste management for the production of materials and biofuel. *Journal of Hazardous Materials Advances*, *10*, 100266. https://doi.org/10.1016/j.hazadv.2023.100266

ASNZS. (2021). *Australian/New Zealand Standard (AS/NZS) 3500.2.2021 Plumbing and Drainage Part 2: Sanitary plumbing and drainage*.

Battistoni, P., Fatone, F., Passacantando, D., & Bolzonella, D. (2007). Application of food waste disposers and alternate cycles process in small-decentralized towns: A case study. *Water Research*, *41*(4), 893-903. https://doi.org/https://doi.org/10.1016/j.watres.2006.11.023

Bolzonella, D., Pavan, P., Battistoni, P., & Cecchi, F. (2003). The under sink garbage grinder: A friendly technology for the environment. *Environmental Technology*, *24*(3), 349-359. https://doi.org/10.1080/09593330309385567

Brisbane City Council. (2024). *Compost rebate program*. Retrieved 15/4 from https://www.brisbane.qld.gov.au/clean-and-green/green-home-and-community/sustainable-gardening/compost-and-food-waste-recycling/compost-rebate-program#:~:text=%24200%20%2D%20for%20the%20purchase%20of,a%20maximum%20rebate%20of%20%2470.

Caldeira, C., De Laurentiis, V., Corrado, S., van Holsteijn, F., & Sala, S. (2019). Quantification of food waste per product group along the food supply chain in the European Union: a mass flow analysis. *Resources, Conservation and Recycling*, *149*, 479-488. https://doi.org/10.1016/j.resconrec.2019.06.011

Carey, C., Phelan, W., & Boland, C. (2008). *Examining the use of food waste disposers*. United States Environmental Protection Agency (US EPA).

Carlu, E., Truong, T., & Kundevski, M. (2019). *Biogas opportunities for Australia*. ENEA Consulting.

Chrisafis, A. (2016, 1/04/2024). French law forbids food waste by supermarkets. *The Guardian*.

City of Melbourne. (2024). *Waste and recycling centre*. https://www.melbourne.vic.gov.au/residents/waste-recycling/Pages/waste-recycling-centre.aspx

Clarke, W. (2022, 16/12/2022). Despite government delays, food waste recycling bins are coming to your kitchen sooner than you think. *The Conversation*. https://theconversation.com/despite-government-delays-food-waste-recycling-bins-are-coming-to-your-kitchen-sooner-than-you-think-195734

Commonwealth of Australia. (2017). *National Food Waste Strategy: Halving Australia's food waste by 2030*. Commonwealth of Australia.

CWWA. (2019). *Residential Food Waste Grinders: Issues Analysis Paper*. Canadian Water and Wastewater Association. https://cwwa.ca/wp-content/uploads/2019/10/Food-Waste-Grinder_WhitePaper.pdf

Davis, B., Graham, A., & Hearn, K. (2005). *Evaluation of food waste disposal units and their part in municipal waste management* 9th European Biosolids Conference,

DAWE. (2020). *National Waste Report 2020*. Department of Agriculture, Water and the Environment.

de Haas, D., Appleby, G., Charakos, G., & Dinesh, N. (2018). Benchmarking energy use for wastewater treatment plants: A summary of the 2015-16 benchmarking study. *Water e-journal: Online Journal of the Australian Water Association*, *3*(2).

de Haas, D. W., Pepperell, C., & Foley, J. (2013). Perspectives on greenhouse gas emission estimates based on Australian wastewater treatment plant operating data. *Water Science and Technology*, *69*(3), 451-463. https://doi.org/10.2166/wst.2013.572

DeOreo, W. B., Mayer, P. W., Martien, L., Hayden, M., Funk, A., Kramer-Duffield, M., Davis, R., Henderson, J., Raucher, B., Gleick, P., & Heberger, M. (2011). *California single family water use efficiency study*. Aquacraft, Inc. Water Engineering Management.

DES, Q. (2022). *Queensland Organics Action Plan 2022–2032*. Department of Environment and Science.

Edwards, J., Othman, M., Crossin, E., & Burn, S. (2018). Life cycle assessment to compare the environmental impact of seven contemporary food waste management systems. *Bioresource Technology*, *248*, 156-173. https://doi.org/10.1016/j.biortech.2017.06.070

EEA. (2021). *Overview of national waste prevention programmes in Europe: Norway*. European Environment Agency. https://www.eea.europa.eu/themes/waste/waste-prevention/countries/2021-waste-prevention-country-profiles/norway-waste-prevention-country-profile-2021.pdf

EEA. (2023a). *Waste prevention country profile: Denmark*. European Environment Agency. https://www.eea.europa.eu/themes/waste/waste-prevention/countries/2023-waste-prevention-country-fact-sheets/denmark_waste_prevention_2023#:~:text=FOOD%20WASTE%20PREVENTION,-Food%20waste%20generation&text=The%20amount%20of%20food%20waste%20from%20households%20has%20decreased%20by,8%20%25%20per%20household%20per%20week.&text=The%20REFOOD%20label%20is%20a,minimising%20and%20recycling%20food%20waste.

EEA. (2023b). *Waste prevention country profile: France*. European Environment Agency. https://www.eea.europa.eu/themes/waste/waste-prevention/countries/2023-waste-prevention-country-fact-sheets/france_waste_prevention_2023

ELWP, V. (2020). *Recycling Victoria: A new economy*. State of Victoria Department of Environment, Land, Water and Planning.

ESC. (2022). *Greenhouse gas co-efficient 2022*. Essential Services Commision. https://www.esc.vic.gov.au/sites/default/files/documents/greenhouse-gas-coefficient-2022.pdf

ESC. (2024a). *Average household water bills in Victoria*. Essential Services Commission. Retrieved 1/3 from https://www.esc.vic.gov.au/water/water-prices-tariffs-and-special-drainage/average-household-water-bills-victoria

ESC. (2024b). *Victorian Default Offer*. Essential Services Commission. Retrieved 1/3 from https://www.esc.vic.gov.au/electricity-and-gas/prices-tariffs-and-benchmarks/victorian-default-offer

Evans, T. D. (2012). Comment on Iacovidou, E.; Ohandja, D.-G. and Voulvoulis, N. (2012) Food waste disposal units in UK households: The need for policy intervention. Science of the Total Environment 423 1–7. *Science of the Total Environment*, *437*, 433-434. https://doi.org/10.1016/j.scitotenv.2012.07.042

Evans, T. D., Andersson, P., Wievegg, Å., & Carlsson, I. (2010). Surahammar: a case study of the impacts of installing food waste disposers in 50% of households. *Water and Environment Journal*, *24*(4), 309-319. https://doi.org/10.1111/j.1747-6593.2010.00238.x

FIAL. (2021). *The National Food Waste Strategy Feasibility Study – Final Report*. Food Innovation Australia Limited.

Graunke, R. E., & Wilkie, A. C. (2014). Examining the mechanisms of short-term solubilization of ground food waste for high-rate anaerobic digestion. *International Biodeterioration & Biodegradation*, *86*, 327-333. https://doi.org/https://doi.org/10.1016/j.ibiod.2013.10.007

Green Industries SA. (2020). *Supporting the circular economy: South Australia's Waste Strategy 2020 - 2025*. Green Industries SA, Government of South Australia.

Guan, J., Zhuang, H., Lau, C. Y., Leng, L., Yeung, C. S., Vuppaladadiyam, A. K., Wang, H., Tse, H.-Y., & Leu, S.-Y. (2022). Energy and carbon footprint analysis of municipal wastewater treatment process integrated with food waste disposer. *Journal of Cleaner Production*, *375*, 134063. https://doi.org/10.1016/j.jclepro.2022.134063

Guo, H., van Lier, J. B., & de Kreuk, M. (2020). Digestibility of waste aerobic granular sludge from a full-scale municipal wastewater treatment system. *Water Research*, *173*, 115617. https://doi.org/10.1016/j.watres.2020.115617

Gustafsson, M., Ammenberg, J., & Murphy, J. D. (2022). *IEA Bioenergy Task 37 – A perspective on the state of the biogas industry from selected member countries*. IEA Bioenergy.

Iacovidou, E., Ohandja, D. G., Gronow, J., & Voulvoulis, N. (2012). The Household Use of Food Waste Disposal Units as a Waste Management Option: A Review. *Critical Reviews in Environmental Science and Technology*, *42*(14), 1485-1508. https://doi.org/10.1080/10643389.2011.556897

Iacovidou, E., Ohandja, D. G., & Voulvoulis, N. (2012). Food waste disposal units in UK households: The need for policy intervention. *Science of the Total Environment*, *423*, 1-7. https://doi.org/10.1016/j.scitotenv.2012.01.048

Kim, D., & Phae, C. (2023). Analysis of the Effect of the Use of Food Waste Disposers on Wastewater Treatment Plant and Greenhouse Gas Emission Characteristics. *Water*, *15*(5), 940. https://www.mdpi.com/2073-4441/15/5/940

LGA. (2015). *The impact of household food waste disposers: Results of the field trials in Shropshire*. Local Government Association.

Liu, C., Hotta, Y., Santo, A., Hengesbaugh, M., Watabe, A., Totoki, Y., Allen, D., & Bengtsson, M. (2016). Food waste in Japan: Trends, current practices and key challenges. *Journal of Cleaner Production*, *133*, 557-564. https://doi.org/https://doi.org/10.1016/j.jclepro.2016.06.026

Lundie, S., & Peters, G. M. (2005). Life cycle assessment of food waste management options. *Journal of Cleaner Production*, *13*(3), 275-286. https://doi.org/https://doi.org/10.1016/j.jclepro.2004.02.020

Marashlian, N., & El-Fadel, M. (2005). The effect of food waste disposers on municipal waste and wastewater management. *Waste Management & Research*, *23*(1), 20-31. https://doi.org/10.1177/0734242x05050078

Melbourne Water. (2024a). *Biogas*. Retrieved 1/4 from https://www.melbournewater.com.au/water-and-environment/energy/biogas

Melbourne Water. (2024b). *Melbourne Water injects millions into Western Treatment Plant upgrade*. Retrieved 23/4 from https://www.melbournewater.com.au/about-us/what-we-do/news/melbourne-water-injects-millions-western-treatment-plant-upgrade-0

Moñino, P., Aguado, D., Barat, R., Jiménez, E., Giménez, J. B., Seco, A., & Ferrer, J. (2017). A new strategy to maximize organic matter valorization in municipalities: Combination of urban wastewater with kitchen food waste and its treatment with AnMBR technology. *Waste Management*, *62*, 274-289. https://doi.org/10.1016/j.wasman.2017.02.006

Moult, J. A., Allan, S. R., Hewitt, C. N., & Berners-Lee, M. (2018). Greenhouse gas emissions of food waste disposal options for UK retailers. *Food Policy*, *77*, 50-58. https://doi.org/10.1016/j.foodpol.2018.04.003

NSW CEA. (2024). *Bioenergy*. NSW Climate and Energy Action. Retrieved 15/4 from https://www.energy.nsw.gov.au/nsw-plans-and-progress/major-state-projects/shift-renewables/bioenergy

NSW DPIE. (2021). *NSW Waste and Sustainable Materials Strategy 2041*. NSW Department of Planning, Industry and Environment.

NYC DEP. (1997). *The Impact of Food Waste Disposers in Combined Sewer Areas of New York City*. NYC Department of Environmental Protection.

Okayama, T., & Watanabe, K. (2024). Performance of the Food Waste Recycling Law in Japan with Reference to SDG 12.3. *Recycling*, *9*(1), 18. https://www.mdpi.com/2313-4321/9/1/18

Ozgun, H., Cicekalan, B., Akdag, Y., Koyuncu, I., & Ozturk, I. (2021). Comparative evaluation of cost for preliminary and tertiary municipal wastewater treatment plants in Istanbul. *Science of the Total Environment*, *778*, 146258. https://doi.org/10.1016/j.scitotenv.2021.146258

PlumbingCosts. (2024). *How much do plumbers charge?* Retrieved 01/04 from https://www.plumbingcosts.com.au/how-much-do-plumbers-charge-1

Randell, P., Pickin, J., & Grant, B. (2014). *Waste Generation and Resource Recovery in Australia*

Department of Sustainability, Environment, Water, Population and Communities.

Sanciolo, P., Rivera, E., Navaratna, D., & Duke, M. C. (2022). Food Waste Diversion from Landfills: A Cost–Benefit Analysis of Existing Technological Solutions Based on Greenhouse Gas Emissions. *Sustainability*, *14*(11), 6753.

Seberini, A. (2020). Economic, social and environmental world impacts of food waste on society and Zero waste as a global approach to their elimination. SHS Web of Conferences,

Shen, Y., Linville, J. L., Urgun-Demirtas, M., Mintz, M. M., & Snyder, S. W. (2015). An overview of biogas production and utilization at full-scale wastewater treatment plants (WWTPs) in the United States: Challenges and opportunities towards energy-neutral WWTPs. *Renewable and Sustainable Energy Reviews*, *50*, 346-362. https://doi.org/10.1016/j.rser.2015.04.129

Sheridan, J., Carey, R., & Candy, S. (2016). *Foodprint Melbourne: What does it take to feed a city?* Victorian Eco Innovation Lab, the University of Melbourne.

Smoczynski, L., Kosobucka, M., Smoczynski, M., Ratnaweera, H., & Pieczulis-Smoczynska, K. (2016). Sizes of particles formed during municipal wastewater treatment. *Water Science and Technology*, *75*(4), 971-977. https://doi.org/10.2166/wst.2016.546

Sydney Water. (2023). *Food waste processing via anaerobic/aerobic digestion: A guide to disposing of food waste as commercial trade wastewater*. SydneyWater. https://www.sydneywater.com.au/content/dam/sydneywater/documents/food-waste-processing-anaerobic-aerobic-digestion.pdf

Thota Radhakrishnan, A. K., van Lier, J. B., & Clemens, F. H. L. R. (2018). Rheological characterisation of concentrated domestic slurry. *Water Research*, *141*, 235-250. https://doi.org/10.1016/j.watres.2018.04.064

UK DEFRA. (2023). *UK statistics on waste*. Department for Environment Food & Rural Affairs. Retrieved 1/4 from https://www.gov.uk/government/statistics/uk-waste-data/uk-statistics-on-waste#biodegradable-municipal-waste-bmw-sent-to-landfill

UK Legislation. (2021). *Environment Act 2021*. https://www.legislation.gov.uk/ukpga/2021/30/section/57/enacted

UNEP. (2021). *Food Waste Index Report 2021*. United Nations Environment Programme.

United Nations General Assembly. (2015). *Transforming our world: the 2030 Agenda for Sustainable Development*. Resolution adopted by the General Assembly on 25 September 2015. Retrieved 1/4/2023 from https://undocs.org/en/A/RES/70/1

US EPA. (2023). *Estimates of generation and management of wasted food in the United States in 2019* United States Environmental Protection Agency. https://www.epa.gov/facts-and-figures-about-materials-waste-and-recycling/food-material-specific-data#:~:text=EPA%20estimates%20that%20in%202019%2C%2066%20million%20tons,this%20waste%20%28about%2060%25%29%20was%20sent%20to%20landfills.

Wainberg, R., Nielse, J., Lundie, S., Peters, G., Ashbolt, N., Russell, D., & Jankelson, C. (2000). *Assessment of Food Disposal Options in Multi-Unit Dwellings in Sydney (2883R)*. CRC for Waste Management and Pullution Control Limited.

Water Corporation. (2024). *Food waste disposal units*. Retrieved 1/4 from https://www.watercorporation.com.au/Help-and-advice/Trade-waste/Trade-waste-in-your-business/Other-industries/Food-waste-disposal-units

Western Australian Legislation. (2013). *Water Services Act 2012: Water Services Regulations 2013*. https://www.legislation.wa.gov.au/legislation/statutes.nsf/main_mrtitle_13116_homepage.html

Wilkinson, K., Price, J., Biala, J., & McDonald, D. (2021). *Review of Regulations and Standards for Recycled Organics in Australia: Final Report for Department of Agriculture, Water and Environment*. Frontier AG & Environment.

WMRR. (2021). *Solving Australia's food waste problem requires purposeful planning*. Waste Management and Resource Recovery Association of Australia. Retrieved 1/4 from https://www.wmrr.asn.au/Web/Media/Media_Release/2021/Solving-Australia-s-food-waste-problem-requires-purposeful-planning-.aspx?WebsiteKey=00969b94-a8ff-49f3-8d36-264248d5094d

WRc. (2010). *National Food Waste Programme (Work Package 1.1) Comparison of the Sustainability of Food Waste Disposal Options*. UC8427.

Yang, X., Okashiro, T., Kuniyasu, K., & Ohmori, H. (2010). Impact of food waste disposers on the generation rate and characteristics of municipal solid waste. *Journal of Material Cycles and Waste Management*, *12*(1), 17-24. https://doi.org/10.1007/s10163-009-0268-y

Zupančič, M., Možic, V., Može, M., Cimerman, F., & Golobič, I. (2022). Current Status and Review of Waste-to-Biogas Conversion for Selected European Countries and Worldwide. *Sustainability*, *14*(3), 1823. https://www.mdpi.com/2071-1050/14/3/1823

APPENDIX A: CALCULATIONS

- **Energy required for truck waste collection**

$$\text{Energy required (kWh kg}^{-1}\text{ food waste)}$$
$$= \text{Truck energy consumption (kWh km}^{-1}) \left[\times \frac{2 \times \text{Distance to transfer station (km)}}{\text{Garbage truck capacity (kg)}} + \frac{2 \times \text{Distance from transfer station to landfill (km)}}{\text{Garbage truck capacity (kg)}}\right]$$

$$\text{Energy required (kWh kg}^{-1}\text{ food waste)} = 3.699 \times \frac{2 \times 5.5}{24000} + 3.699 \times \frac{2 \times 27.2}{24000}$$
$$= 0.01 \text{ kWh kg}^{-1} \text{ food waste}$$

where truck energy consumption is 3.699 kWh/km and garbage truck capacity is 24 Ton (Sanciolo et al., 2022). It is assumed that truck routes are optimized for maximum collection.

- **Equivalent Carbon Dioxide (CO_2e) generated from truck waste collection**

$$CO_2\text{e generated (kg } CO_2\text{e kg}^{-1}\text{ food waste)}$$
$$= \text{Truck } CO_2\text{e (kg}CO_2\text{e km}^{-1}\text{ kg}^{-1}\text{of food waste)} \times \text{Distance travelled (km)}$$
$$= 0.0002 \times 34.6 = 0.0069 \text{ kg } CO_2\text{e kg}^{-1} \text{ food waste}$$

where the truck CO2e is 0.0002 as reported by Moult et al. (2018).

- **Energy required for truck waste collection**

$$\text{Energy required (kWh kg}^{-1}\text{ food waste)}$$
$$= [\text{Energy required for COD removal} - \text{Energy recovered by COD removal}]$$
$$= 0.35 \text{ kgCOD kg}^{-1} \text{ food waste}[2.8 \text{ kWh kg}^{-1}\text{COD} - 1.3 \text{ kWh kg}^{-1}\text{COD} - 1.3]$$
$$= 0.53 \text{ kWh kg}^{-1} \text{ food waste}$$

- **Equivalent CO_2e generated from WWTP**

$$CO_2\text{e generated (kg } CO_2\text{e kg}^{-1}\text{ food waste)}$$
$$= \text{green house gas coefficient (kg } CO_2\text{e kWh}^{-1})$$
$$\times \text{ Energy required (kWh kg}^{-1}\text{ food waste)} = 0.53 \times 1.06$$
$$= 0.56 \text{ kg } CO_2\text{e kg}^{-1} \text{ food waste}$$

- **Capital cost for tertiary treatment with digester**

$$\text{Capital cost (AUD\$)} = 15812.5\ Q^{0.7546}$$

where Q is the wastewater discharge in m^3/day.

- **Cost for tertiary treatment operation and maintenance with digester**

$$\text{Capital cost (AUD\$)} = 22860.75\ Q^{0.5008}$$

where Q is the wastewater discharge in m^3/day.

Appendix B: Food sample weights

Table B1. Standard load food sample weights for Grinding Rate test

Food weight (g)	WDU-3ASAU			WDU-4ASAU		
	1	2	3	4	5	6
Yellow Banana Skin	149.29	150.87	147.94	151.23	145.47	145.37
Corn cobs	98.52	97.94	99.62	98.76	99.71	100.65
Grapefruit rind	95.35	96.54	96.36	98.28	97.7	96.92
Celery	50.72	49.60	48.88	50.06	48.46	50.04
Dry paper towels + tea bags (dry)	8.13	8.13	8.13	8.13	8.13	8.13
Dry paper towels + tea bags (wet)	18.37	19.28	18.71	19.54	21.17	18.29
Potatoes	189.92	190.00	190.20	190.89	189.15	189.49
Mass of load (m_L)	602.17	604.23	601.71	608.76	601.66	600.76

Table B2. Abey load food sample weights for Grinding Rate test

Food weight (g)	WDU-3ASAU			WDU-4ASAU		
	1	2	3	4	5	6
Meat Scraps	100.71	100.68	86.00	100.35	100.79	99.72
Apple cores (2)	44.81	51.27	46.89	47.21	18.23	29.22
Eggshells (2)	21.93	14.15	13.45	13.92	14.7	13.63
Melon rinds	100.13	101.75	101.25	99.44	99.35	101.66
Pistachio shells	30.46	30.53	30.00	29.85	29.37	30.11
Whole cooked prawns (2)	59.52	58.32	46.81	57.64	56.88	59.58
Mass of load (m_L)	318.56	289.95	303.81	327.10	326.17	294.4

Table B3. Mix load food sample weights for the Degree of Fineness

Food weight (g)	WDU-3ASAU		WDU-4ASAU	
	1	2	3	4
Rib bones	123.55	121.41	126.31	128.05
Carrot	125.65	125.35	125.88	125.64
Celery	125.90	126.11	124.33	126.27
Lettuce	126.07	125.41	125.08	125.90
Total	501.17	498.28	501.60	505.86